SILVICULTURAL TOPICS
CHAPTER THREE

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LINKAGE BETWEEN RIPARIAN BUFFER FEATURES AND REGENERATION, BENTHIC COMMUNITIES AND WATER TEMPERATURE IN HEADWATER STREAMS, WESTERN OREGON

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ABSTRACT

Riparian forests can be managed using a range of harvesting and regeneration methods to achieve multiple environmental and economic objectives. In this study, seven low-elevation second- or third-order streams were subjected to either patch clearcutting with no buffers or one-sided narrow buffers divided by uncut reaches. Of these streams, four were sites of intensive regeneration experiments, and the other three evaluated only the effect of harvest pattern on water temperature. Regeneration was successfully installed along four streams with intensive planting experiments in which three clearcuts on each spanned the stream for distances of 90 or 180 m. Regeneration cutting in these drainages included clearcutting to the water's edge in openings amounting to 25% of 1,500-m reaches. Planting tests evaluated three species: Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), and western redcedar (*Thuja plicata* Donn). Douglas-fir was represented by two different stock types, bareroot plug+1 and 1+1 transplants on two sites and bareroot 1+1 and 2+0 seedlings on the other two sites. All regeneration efforts are on a path that will eventually result in conifer-dominated riparian zones, with degree of success influenced by choice of stock type, overtopping cover, animal damage and frost. Damage from deer (*Odocoilius hemionus columbianus* Rich.), elk (*Cervus elaphus canadensis* L.), beavers (*Castor canadensis* Kuhl.) and/or mountain beavers (*Aplodontia rufa* Raf.) occurred on all four stream systems. There were no patterns of beaver or mountain beaver damage among the stock types; browsing on hemlock was minor. Growth status of the plantations at age four was a function of overtopping vegetation where there is low risk of frost damage; freezing temperatures were was limiting to seedling growth along one stream. All three of the primary tree regeneration species studied exhibited decreased growth with overtopping. As has been found on upland sites elsewhere, size of seedlings had a strong influence on their competitive ability in riparian zones.

Aquatic insect communities displayed approximately a doubling of abundance with either unbuffered clearcuts or those buffered on just the south side. Minor shifts occurred in insect community composition; none decreased and certain genera increased markedly, with a net increase in abundance of food organisms that could be utilized by salmonids. Stream temperature increased in response to silvicultural clearing when all cover was removed. Warming and cooling trends on the four streams with regeneration experiments were not definitive, but were within the range reported by Zwieniecki and Newton (1999). In the three clearcuts along streams having narrow buffer screens on the south side only, no evidence of warming was observed in uncut areas 300 m downstream of cut areas compared with pre-harvest temperature patterns.

KEYWORDS: Stream temperature, reforestation, aquatic macroinvertebrates, riparian management, buffer design.

INTRODUCTION

Some states have adopted regulatory best management practices (BMPs) while other states use voluntary BMPs to address the interface between streams and forests to protect water quality and maintain aquatic habitat. These state BMPs must be sensitive to forest landowner needs to ensure that the practices will benefit fishery resources, meet other environmental objectives, and maintain landowner support and interest in forest management (Hairston-Strang and

Adams 1997). Reports of potentially harmful elevations of stream temperatures in some studies have raised concerns about harvesting practices (Brown and Krygier 1970, Caldwell et al. 1991), leading to eventual adoption of prescriptive BMPs. Although current riparian regulations in Oregon allow for some harvesting within riparian areas, such management is expensive and may not eventually be appropriate for regeneration of desired species or for providing economic benefits to justify it. Berg (1995) explored the notion that active management of riparian forests can be both economically and ecologically beneficial. His analysis relies on partial cutting of riparian stands and model projections of growth. In this report, we explore several physical and biological interactions associated with clearcut harvesting adjacent to streams with a goal of providing long-term renewal of desirable species while protecting streams.

Removal of riparian forest cover immediately over streams can increase solar radiation and stream temperature in the short term (Brown and Krygier 1970, Tait et al. 1994). Even if vegetation removal is such that there are no direct effects on fish, fish may be impacted by indirect effects on aquatic invertebrates. Fish feed intensively on aquatic invertebrates as both nymphs and adults. Immature benthic (bottom-dwelling) insects live primarily on in-stream substrates but actively and passively enter the water flow and drift downstream, becoming highly vulnerable to predation by fish. Temperature and solar radiation loads may affect insects by altering food sources, life cycles, and habitat (Newbold et al. 1980, Murphy and Hall 1981, Tait et al. 1994), with potential changes being either negative or positive, depending on organisms used for a response index. Changes in stream temperature directly impact numbers, fecundity, body size emergence phenology, metabolic rate and diurnal patterns of insects (Merritt and Cummins 1984). Temperature changes indirectly affect aquatic insects through oxygen saturation levels, respiration rates, food availability and nutritive value, and competition from other insects (Sweeney 1984; Walsh 1996). Photosynthetically-active radiation levels affect the amount of primary production in the stream, and thus, influence the kind and amount of food for benthic (bottom-dwelling) insects. Changes in some species may occur independently of others, so relative abundance of species as well as total species present are both important indicators of change. High levels of primary productivity can be achieved with high levels of diffuse radiation, whereas most energy leading to increased stream-water temperature is attributed to direct radiation (Brown 1969). In short, there are several elements of riparian habitat management where there can be legitimate debate about whether a particular procedure enhances one element of the stream environment or degrades another, and whether practices with consistent improvement in over-all fish habitat are achievable.

Whether any given level of protection is needed depends on status of resources to be protected. Of particular interest for many forest settings is the interaction of water temperature and food supply for fish (Sullivan et al. 2000). States evaluate stream water quality conditions using standards that include temperature. Whether the desired temperature criteria can be met with buffer design depends on whether any given stream is in a desirable range of temperatures to start with, and whether forest management approaches affect water quality just locally versus downstream. The studies we undertook were designed to provide coarse-resolution answers to questions about managing for favorable temperature and food supply while also setting up future conifer-dominated stands for recruitment of large wood and other values.

We begin with several assumptions. First, that findings of Warren (1971) and Sullivan et al. (2000) relating to interaction of food supply and tolerance of elevated temperatures are valid, and that short-term exposures of well-fed fish to temperatures above 17.8°C (64°F) will not materially harm most salmonids. Second, that future structural features of streams will be enhanced by long-term development of durable conifers suitable for either tree-fall or placement in streams within the riparian management areas. Finally, that abundance and diversity of benthic macroinvertebrates are indicators of potential food supply for fish in the presence of adequate pool and riffle habitat and moderate temperatures (Wilzbach et al. 1988).

This report summarizes data from a series of experiments that integrates silvicultural manipulation of riparian forests with three key elements of riparian habitat: future riparian stand establishment, water temperature, and aquatic macroinvertebrate abundance. These experiments evaluate a series of reforestation techniques based on upland experience to facilitate rapid establishment of conifers and green-up of riparian systems following commercial harvest. They were based on two approaches to clearcut harvesting: clearcutting to the water's edge on both sides versus clearcutting with one-sided buffers. Harvests with no buffers were combined with trials of several coniferous stock types with and without weed control with the general approach outlined by Newton et al. (1993). The object of the one-sided buffer was to manipulate streamside shade to gain protection from direct radiation while allowing maximum regeneration opportunity. The regeneration studies rely on repeated measurements of several planted coniferous tree

species and competing hardwood and shrub cover, measurement of water temperatures along streams passing through the various treatments (including data from Zwieniecki and Newton 1999), and samplings of bottom-dwelling macroinvertebrates to assess impacts on potential abundance of aquatic insects well utilized by salmonids.

Our specific objectives were to: 1) evaluate the regeneration success and growth of combinations of planting stock species and size combined with and without overstory removal and shrub control in streamside environments, 2) determine how alternative approaches to removing tree cover near streams influence abundance and diversity of benthic insect communities, and 3) develop a coarse-resolution estimate of whether openings to facilitate forest rehabilitation affect downstream water temperature. Robison and Beschta (1990) and others have reported that most large wood contributing to stream structure originates within 15 m from the banks. The first objective specifically addresses the problem of establishing conifers near streams on low-elevation sites in the Douglas-fir (*Pseudotsuga menziesi*i (Mirb.) Franco var. *menziesii*) Region where most low-elevation streams are bounded largely by hardwoods. The persistence of these species as functional large wood in streams tends to be short compared to that of large conifers. While there are reports that upslope or upstream sources may provide much large wood for these streams (Reeves et al. 2003), a local source is presumably a better guarantee that a local supply of wood will be present when needed. Thus, in the event incentives develop for establishing large conifers, there is a need to understand the environments and management approaches in which such trees may develop. The other objectives address impacts of buffer design on aspects of the stream system that might relate to fish.

METHODS

Layout

The studies occurred in two primary phases, both of which utilized fish-bearing headwaters streams passing through low-elevation, intensively managed forest ownerships. Precipitation for all creeks is between 1200 and 2000 mm/year, of which roughly 85 percent occurs between October and April. In Phase I, four second- or third-order fish-bearing streams (Ames, Bark, Buttermilk, Mosby Creeks, fig. 1) were scheduled for clearcut harvests, with coniferous reforestation according to hardwood conversion rules (Oregon Forest Practices Act) that allowed clearcutting to the streambank according to a written plan. These were the streams on which all reforestation experiments were conducted.
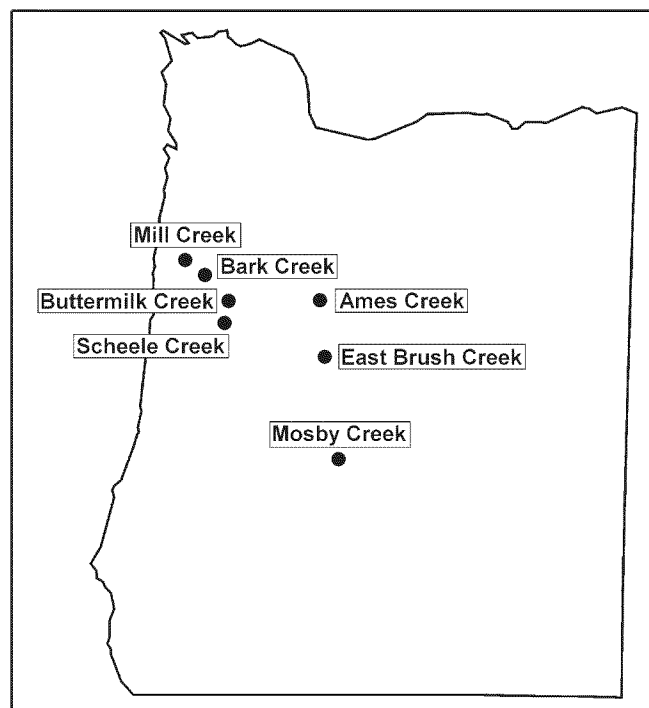

Figure 1—Map of stream locations in western Oregon.

Each Phase I stream was laid out to provide a basis for both reforestation and study of effects of small clearcuts with no buffers on water temperature. In each, a 1500-m reach was identified for study. Two creeks (Bark and Buttermilk) are in the central part of the Oregon Coast Range, and are bedded in sedimentary rock. Buttermilk Creek drains a basin of 702 ha at a gradient of three percent, and Bark Creek drains 1560 ha with a gradient of less than two percent in the study reach. The other two (Ames and Mosby) drain lower slopes of the Oregon Cascade Mountains, and are bedded in basalt bedrock and boulders. Ames Creek drains a basin area of 1441 ha and has a gradient in the study reach of three percent. Mosby Creek drains 26,022 ha with a gradient of five percent in the study reach.

Buttermilk and Bark Creeks flow between terraces largely of deposits of medium-textured material from sedimentary rocks of the Tyee formation, a readily weathered siltstone in this area. These soils are extremely productive for either conifers or red alder (*Alnus rubra* Bong.). Except in poorly drained depressions and gravel piles, all soils are of good to excellent productivity, capable of growing a Douglas-fir tree 38m to 40 m tall in 50 years (King 1966). Stands removed in the harvests were dominated by red alder, with understories of salmonberry (*Rubus spectabilis* Pursh.). Ames Creek soils are a mixture of alluvial gravelly loam terraces with scattered wet depressions, and residual

well drained Jory silty clay loam soils; site quality is heterogeneous, and estimated 50-year Douglas-fir site index ranges from 33 m to 37 m (King 1996). The previous stands were primarily hardwoods, mixtures of bigleaf maple (*Acer macrophyllum* Pursh.) and black cottonwood (*Populus trichocarpa* T. & G.) with some willows (*Salix* [Tourn.] L.) and Douglas-fir. Of the four Phase I streams, Mosby Creek differed markedly from the others in having been subjected to placer mining, probably more than 50 years previously. The soil there was largely comprised of a thin layer of organic litter on top of piles of gravel and boulders, with occasional patches of original gravelly loam soil. Planting was exceedingly difficult. The stand removed in preparation for this study consisted primarily of immature red alder with intermixed conifers of assorted sizes and understories of mixed shrubs, depending on degree of suppression. Estimated 50-year site index for Douglas-fir at Mosby Creek ranges from 30 m to 35 m (King 1996).

The 1500-m study reach for each of the four Phase I streams was divided in half so that one 750-m sub-reach had 25% of its length clearcut-harvested in one 180-m reach and the other sub-reach had two 90-m openings (fig. 2). The remainder of each 1500-m reach was uncut or remained buffered according to pre-1994 Oregon Forest Practice Rules (22 m or more both sides), and there was at least 200 m of uncut stand or standard buffer between any two clearcuts. Logging of all units was completed in late 1992. Phase I included all reforestation experiments, for which all clearcuts were made with no buffers.

Phase II streams evaluated only the influence of a one-sided buffer design on water temperature and stream productivity. Each had a single clearcut installation, in late 1993, with a buffer represented by a "sun-sided" vegetative screen on each of three similar-sized streams (East Brush, Mill, and Scheele Creeks, figs. 1 and 3). East Brush Creek drains about 1100 ha of basaltic foothills of the Cascade Mountains. In the Coast Range, Scheele Creek drains 1743 ha of deeply weathered Siletz River basaltic rock, and is bounded by Jory soil with negligible terrace areas. Mill Creek drains about 440 ha of sedimentary rocks and deep soils. In each, cutting was done on both sides of the stream so that a 760m to 800-m reach was exposed, but a vegetated screen 12-m wide remained between any point in the stream and the path of direct solar radiation through the summer. The screen consisted of whatever trees and shrubs were present in the uncut condition so long as they were within 12 m of the streambank. Where streams were east-west in orientation, no buffer was left on the north side, but the sun shining on open water between 9:00 AM and 6:00 PM PDT was intercepted to a major degree by a screen of
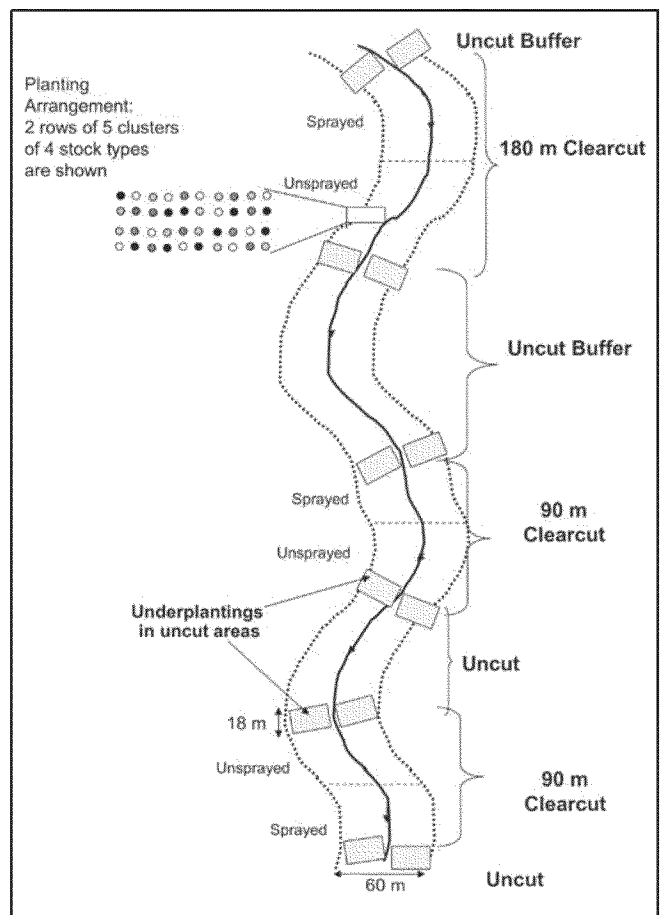

Figure 2—Schematic of unit layout for reforestation (Phase I) studies.

cover. The protection was provided in meandering streams by leaving all cover in a 150-degree fan-shaped range of directions between true azimuths of 120 and 270 degrees for a distance of 12 m from the bank as shown in figure 3 (adapted from Newton 1993).

All cutting units were laid out specifically for the study. The methods used in these experiments were within the bounds of Oregon Forest Practice Rules, with the exception that buffer removal to the water's edge and one-sided buffers required specific written plans and research protocols. Logging operations were conducted with equipment and approaches according to cable or ground-based systems normal for each owner with special priority for avoidance of damage to streambanks.

Reforestation

In each clearcut established for Phase I, planting was done with four stock types, including large and small Douglas-fir, and plug+1 western redcedar (*Tsuja plicata* Donn) and western hemlock (*Tsuga heterophylla* (Raf.)
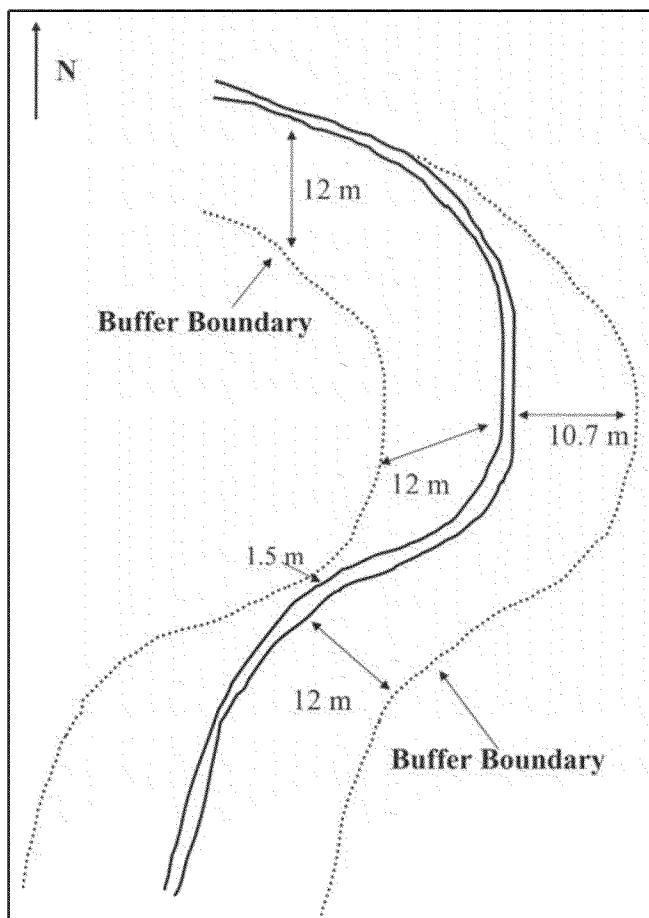
Figure 3—Schematic of fan-projection method of implementing one-sided buffer boundaries for fish-bearing streams.

Sarg.). The seedlings were planted in clusters of four seedlings, one of each species or stock type, in five rows of clusters 6 m apart parallel to the stream, beginning about three m from the bank, overall approximating a 3 x 3-m spacing in a 30-m strip on each side of the stream (fig. 2). The rows extended 18 m into the uncut timber upstream and downstream from each clearcut. Half of each planting was treated with a broadcast site preparation application of glyphosate herbicide so as to leave a 3-m wide untreated zone along the stream at Bark and Buttermilk Creeks. Spot-treated release applications of glyphosate were applied at Ames and Mosby Creeks. For each cutting unit, spraying left a 3-m buffer along both banks of the stream. Spraying extended 20 m into understories of the uncut stands at the sprayed end of the unit where seedlings were to be planted. A total of some 12,000 seedlings were planted. For each seedling, annual measurements during the first four years recorded seedling diameter 15 cm aboveground, diameter at breast height (137 cm, dbh) if seedlings were of sufficient

size, and current and previous-year's height (for confirmation). If data did not match previous year's data, the site was revisited and measurements confirmed.

At each seedling, visual estimates were made of competing cover of herbaceous plants, shrubs, and ferns within a 1-m radius for years one and two, and of overtopping (Howard and Newton 1984) by shrubs and residual trees for all years; residual trees were present primarily in under-planted stands at the ends of each unit. Condition of seedlings, including animal damage from beavers (*Castor canadensis* Kuhl.), mountain beaver (Aplodontia rufa Raf.), and deer (*Odocoilius hemionus columbianus* Rich.)/elk (*Cervus elaphus canadensis* L.) and mortality, were recorded separately.

Stock quality problems were encountered for the Buttermilk and Bark Creek plantations, both of which had ideal conditions for seedling survival and vigor; the same lot of seedlings planted on good sites elsewhere also performed poorly. Survival was lower than expected (65%) at Buttermilk Creek, but we determined that would be acceptable. Survival at Bark Creek was less than 50%, and we replanted there and removed the original survivors. Thus, for Bark Creek, we have only three years of measurement. Freezing damage and severe elk browsing occurred at Mosby Creek. Survival was below 40% after two years, and the landowner elected to replant vacant spots with a mixture of valley ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) 2+0 seedlings, grand fir (*Abies grandis* (Dougl.) Lindl.) plug+1 transplants, western white pine (*Pinus monticola* Dougl.) 1+0 containerized seedlings and western redcedar plug+1 transplants with 5-cm Vexar®, tubes installed before planting. This plantation (replants only) was two years old at last measurement, and the residual seedlings of the original planting reflected four growing seasons under severe elk pressure. They are summarized separately.

For the seedling data, two types of analyses were performed. The first set of analyses examined the relationship between 1) seedlings planted in the clearcut areas and those planted under the residual stands, 2) seedlings in weeded and unweeded areas in the clearcuts, 3) browsed and unbrowsed seedlings in the clearcuts, and 4) browsed and unbrowsed seedlings in the weeded and unweeded areas in the clearcuts. All stock types were analyzed separately, and only those seedlings that were alive at the time of the last measurement were included in the analyses. For these analyses, we used PROC MIXED in SAS® and the analysis of covariance for comparison of regression lines as outlined in Littell et al. (1996). The dependent variable was

stem volume index (an easily measured estimator of total stem volume), calculated as height*basal diameter^2* /12 and the independent variable was year of measurement (zero to three or four). For the second set of analyses, equations for each species and site were developed relating stem volume index in year three (Bark Creek) or four (other sites) to different cover variables. For all species and sites, the best equation was ln(volume index) = β^0+ β^1 ln(initial volume) + β^2 overtopping + β^3 overstory cover. For Bark Creek cover, overtopping and overstory cover were from year three, and for all other sites, from year four.

Benthic Sampling

The sampling design for insects was described by Walsh (1996) and was first developed on the Phase I streams, then extended to Phase II streams. Among the four Phase I streams, Bark Creek was not sampled, because the bottom was covered with fine sediments and organic detritus on which few insects were observed, and the invertebrate biomass was dominated by snails. Initial insect sampling was done on Phase I streams in the first season following harvest (1993), and was repeated in the second year to determine whether populations were still responding. Phase II streams were sampled in July and September, 1995, two years after harvest, and at this time Phase I streams were re-sampled so the two groups of streams could be compared. For both Phases we placed four sampling stations on each stream, with one station upstream at least 20 m from cutting units, one 20+m below the cutting unit, and two within the cutting unit representing the upstream half and downstream half of the cutting unit. All samples were taken in riffles to minimize variance so as to increase sensitivity to harvest effects. In each designated sampling area, all stretches of riffle at least 4.5-m long were noted, and one was randomly chosen for sampling. Fish also consume insects washed down into pools, eddies, backwaters, and the downstream edge of riffles; our sampling system did not capture these drift organisms, and they are not reflected in our findings.

For each sampling station, a 4.5-m length of stream was divided into 45 sample spots in a 3 x 15 grid pattern, each being 1/3 of the channel width wide by 30 cm long (fig. 4). Six of the sections were then sampled each date following a systematic non-aligned design so that two samples were randomly chosen from the 15 sections on each of left, middle, and right part of the stream on each sampling date. These six subsamples were pooled for analysis. Sections of the grid that were sampled in July were not resampled in September.
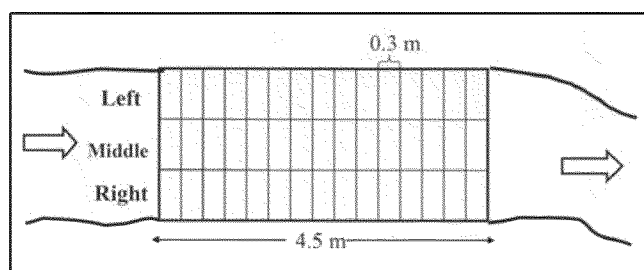

Figure 4—Schematic design of sample location grid used to randomize collections at each benthic sampling station. Buffer design places 12 m of cover between stream and water between 9:00 am and 6:00 pm PDT (azimuth of 120° to 270°).

A 30-cm X 30-cm Surber sample (sample collected from a net into a jar) was collected on the stream bottom within each sampled section with the following specifications. Every rock in the sample square to a depth of 8 cm was cleaned while in the water upstream from the collection net. Variability between subsamples was examined by Walsh (1996) and provided the basis for pooling and sub-sampling for eventual estimation of populations with 50% of the original combined sample material. July samples were sub-sampled again in the lab, to reduce identification time. Identification of insects smaller than 1.1 mm was extremely time-consuming and inexact, hence samples of record were restricted to insects larger than 1.1 mm after the first year. This resulted in small sample sizes, and seasonal comparisons were not attempted. Insects were identified down to the family and genus level where possible. It was extremely difficult to differentiate between genera in some families (e.g., Plecoptera, Perlidae), and a few of the insects were damaged from handling. Genera were assigned to functional feeding groups as per Merritt and Cummins (1994).

Data were analyzed for Phase I in 1993 and 1994 by Walsh (1996) in substantial detail to show immediate responses to clearcutting. Those analyses included only the three streams with reforestation experiments on them. Briefly, influence of substrate composition, stream velocity, temperature and estimated radiation load (evaluated by estimating vegetative cover with a densiometer) were evaluated for their effects on both abundance and diversity of genera, using regressions based on log-transformed insect counts. Residuals from regression were subjected to repeated measures analyses of variance for tests of differences between streams, stations within streams, and interactions.

In 1995, additional samples (third year) were run in Phase I streams and compared to second-year samples in Phase II streams. In view of the increased sample sizes

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when analyzing 1995 samplings of Phase I and II installations together, a separate analysis was used with the primary objective of comparing effects of treatment on large insect populations. Sample data were first transformed to approximate normality in order to run parametric statistics. Analyses of variance (ANOVAs) were performed for total abundance, relative abundance, functional feeding groups, and generic richness for all experimental streams together combining Phase I and II experiments. Means were compared between treatment streams, cut and uncut reaches, and stations. Relative abundance was calculated as the percent of each order in each sample collected. Tukey's Honest Significant Difference test (HSD) was used to compare multiple means. Spearman's rank correlations were done for each of the insect variables with stream temperature and discharge.

Stream Temperature

Stream temperatures were recorded for one year prior and two years immediately following harvest treatments (Phase I) and one year before and one year after for Phase II installations. Four to six thermistors (two-channel Omnidata Datapod Digital Recorders) were placed in each Phase I stream to record water and air temperature, including one or two thermistors above the entire study reach, below the study, in one clearcut, and at one or more other locations in covered sections between clearcuts. The thermistors recorded diurnal temperature fluctuations through the June-September season. Reflecting the importance of temperature extremes, our findings are presented in terms of the influence of treatment on the means of the maximum temperatures occurring in the seven consecutive days with the highest seven-day-mean temperature in each year of record, the standard regulatory criterion. We bring in data from Zwieniecki and Newton (1999) reflecting temperature patterns before and after harvest in sixteen headwaters streams that include the three Phase II installations with sun-sided cover screens. On the other 13 streams they described, clearcut harvesting had been done in which a portion were buffered according to conventional harvest buffer rules (15-21-m buffers on both sides), and half were done with hardwood conversion rules (6-m no-touch buffers, with up to 180 m of stream length per unit having all trees removed).

RESULTS

Reforestation

Each stream had unique features and events that force a presentation of data separately for each. Bark and Buttermilk Creeks, our Coast Range sites, had heavy mortality immediately after planting of all stock types. Because of the replanting at Bark Creek, we only have three years of data for those seedlings. In both Bark and Buttermilk Creek studies, it will also be noted that the "large" and "small" Douglas-fir seedlings were of similar size, but one was 1+1 and the other was plug+1 in nursery regimes, so differences between stock types were minor initially. Ames Creek and Mosby Creek differed from the Coast Range streams in having been prepared too late for site-preparation herbicides, necessitating the use of directed spot herbicide treatments instead. Ames Creek had significant herbicide injury to a number of seedlings. Unrelated to this, Mosby Creek plantings were devastated by frost, elk and rocky substrate resulting from prior mining activity; hence there is great uncertainty in evaluating their future growth.

Bark Creek: Although the large and small Douglas-fir stock types were similar in size when planted, analyses indicate that there was a difference in the volume trajectory ($p = 0.0495$). Examination of the means for year three (table 1) indicate that the small Douglas-fir continued to be smaller than the large Douglas-fir when underplanted in unweeded areas. The volume growth from year two to three (not shown) was three times greater for the large Douglas-fir than the small Douglas-fir in these areas. Results for the two stock types elsewhere along Bark Creek were similar (table 1). Both stock types of Douglas-fir and redcedar were larger ($p < 0.0022$) in clearcut areas than in underplanted areas.

Within the clearcut areas, the large Douglas-fir and western hemlock did not show an increase in growth due to weeding ($p = 0.5041$ and 0.1395, respectively). Spraying did not consistently result in the low levels of cover that are usually needed for major increases in growth (table 2). Redcedar ($p = 0.0024$) and the small Douglas-fir ($p = 0.0137$) did have greater growth in the weeded areas. Year three stem volume for all species was negatively correlated with both overstory cover and overtopping with similar patterns to those shown for Ames Creek in figures 5 to 8.

Buttermilk Creek: Although initial size of the Douglas-fir stock types did not differ, by the end of the fourth year, there were differences ($p < 0.0001$) among the two stock types especially where weeded (table 1). It is not known if these differences are attributable to greater vigor at the time of planting, differences in root mass, or both. However, by the end of the fourth growing season, the large Douglas-fir (1+1) were over 35% larger in volume than the small Douglas-fir (plug+1) in both the weeded and unweeded clearcut areas. All species had greater volume in clearcut areas than in underplanted areas ($p < 0.0724$). All stock types were affected by competition ($p < 0.0619$). In the

Table 1—Height, basal diameter at 15 cm, and stem volume index for year 3 (Bark Creek) and year 4 (Buttermilk, Ames and Mosby Creeks)

		Bark Creek			Buttermilk Creek			Ames Creek			Mosby Creek		
		Height (cm)	Diameter (mm)	Volume (cm^3)	Height (cm)	Diameter (mm)	Volume (cm^3)	Height (cm)	Diameter (mm)	Volume (cm^3)	Height (cm)	Diameter (mm)	Volume (cm^3)
Overstory <=15%													
Unweeded	Large Douglas-fir	130	16.0	115	205	26.2	477	191	22.9	360	137	21.3	198
	Small Douglas-fir	126	15.3	101	167	21.5	298	120	14.8	103	90	14.2	68
	Redcedar	69	9.9	24	147	19.4	234	130	15.7	120	75	11.5	38
	Hemlock	106	10.1	42	176	16.3	182	117	11.0	60	100	11.8	60
Weeded	Large Douglas-fir	120	18.2	129	216	28.8	628	185	24.5	390	126	21.4	188
	Small Douglas-fir	125	18.2	124	179	23.2	356	120	16.5	132	90	15.0	68
	Redcedar	72	11.6	30	164	23.7	378	130	17.8	167	80	13.8	61
	Hemlock	107	11.3	46	205	20.2	326	138	13.8	105	112	13.9	81
Overstory >15%													
Unweeded	Large Douglas-fir	111	12.6	59	168	21.3	293	133	15.3	98	121	24.1	131
	Small Douglas-fir	98	11.4	36	177	20.0	253	85	9.0	25	87	10.7	32
	Redcedar	60	8.1	12	138	17.8	197	99	9.6	31	62	7.3	11
	Hemlock	100	9.0	28	159	14.9	171	71	6.8	11	86	8.8	20
Weeded	Large Douglas-fir	117	14.9	84	204	25.1	409	136	15.6	106	103	13.6	60
	Small Douglas-fir	123	14.8	82	150	18.1	158	93	10.6	41	77	10.4	35
	Redcedar	74	9.9	23	121	14.7	98	108	12.4	60	76	8.4	21
	Hemlock	130	11.4	54	158	13.2	117	100	9.8	34	96	10.3	34

Table 2—Percent overtopping and overstory cover in year 3 (Bark Creek) and year 4 (other creeks) and percent total cover in years 1 and 2 for all creeks in Phase I

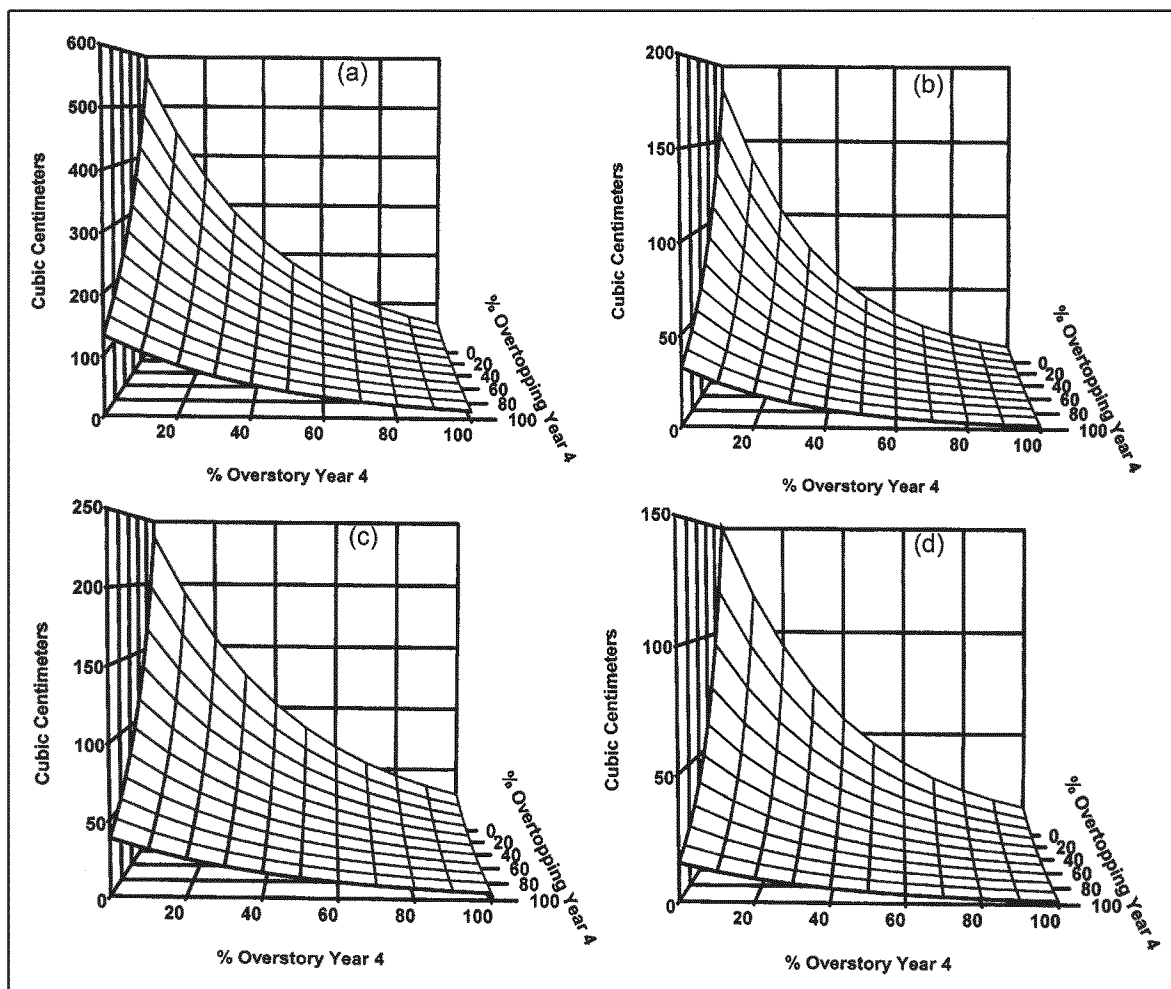|		Bark Creek				Buttermilk Creek				Ames Creek				Mosby Creek			
		Over-topping yr 3	Over-story cover yr 3	Total cover yr 1	Total cover yr 2	Over-topping yr 4	Over-story cover yr 4	Total cover yr 1	Total cover yr 2	Over-topping yr 4	Over-story cover yr 4	Over-cover yr 1	Total cover yr 2	Total topping yr 4	Over-story cover yr 4	Total cover yr 1	Total cover yr 2
													Percent				
Overstory <=15%																	
Unweeded	Large Douglas-fir	21	10	75	90	12	9	76	75	33	11	45	73	19	11	41	61
	Small Douglas-fir	22	11	77	89	16	12	73	75	43	11	40	74	20	9	38	58
	Redcedar	39	12	79	94	24	15	77	76	42	12	46	73	23	10	41	61
	Hemlock	33	10	75	92	23	9	74	77	42	12	42	71	25	8	38	6
Weeded	Large Douglas-fir	3	10	63	76	17	5	51	70	30	12	34	62	10	11	35	55
	Small Douglas-fir	5	10	63	75	20	6	46	68	39	12	32	60	9	9	33	53
	Redcedar	10	12	64	77	25	7	53	70	36	13	33	61	13	15	32	55
	Hemlock	7	10	65	80	23	5	50	67	38	12	35	61	17	13	35	56
Overstory >15%																	
Unweeded	Large Douglas-fir	34	55	70	88	13	48	76	67	36	52	33	66	8	56	20	37
	Small Douglas-fir	47	53	74	89	22	36	82	75	48	43	35	76	17	47	30	35
	Redcedar	58	57	81	98	23	48	75	76	41	51	35	71	14	58	30	29
	Hemlock	45	46	78	95	23	48	72	66	62	42	40	71	4	54	15	31
Weeded	Large Douglas-fir	13	51	54	65	17	28	33	63	14	57	23	53	9	71	31	36
	Small Douglas-fir	11	51	59	65	27	26	36	67	15	59	24	47	7	59	26	30
	Redcedar	11	57	60	71	28	41	48	66	26	57	33	53	4	72	30	32
	Hemlock	11	54	63	74	32	32	37	60	17	54	25	55	24	44	22	39

Figure 5—Stem volume index of (a) unbrowsed large Douglas-fir, (b) unbrowsed small Douglas-fir, (c) unbrowsed western redcedar, and (d) unbrowsed western hemlock at age 4 as a function of overtopping by hardwoods and shrubs, Ames Creek. Note: Scale for z axes is different for each species.

clearcut areas, stem volume of weeded redcedar, hemlock, and small and large Douglas-fir averaged 62, 79, 20, and 32% larger than the unweeded seedlings after four years.

Ames Creek: Differences between stock types (1+1 and 2+0 bare-root) of Douglas-fir were still apparent after four years (table 1). The relative difference between the types has been decreasing, but absolute differences are increasing. Initially, the large 1+1 seedlings were seven times larger in volume than the small 2+0 seedlings. At age four, large seedlings were only three times larger, but had more absolute growth for the first four years.

This installation was established too late to complete broadcast site preparation sprays in the appropriate season, and careful spot treatments during active conifer growth were all that could be done. In the process some seedlings were damaged despite care. There were no differences in growth between weeded and unweeded areas for large Douglas-fir seedlings ($p = 0.2034$), but the other stock types had greater growth in the weeded clearcut areas ($p < 0.0322$). Redcedar and hemlock seedlings averaged 39 and 75% larger in volume, respectively, in weeded clearcut areas than in the unweeded clearcut areas. Regression equations indicated that for all stock types, level of competition affected growth (fig. 5).

Mosby Creek: A variety of sources of damage have severely compromised the initial plantation at Mosby Creek. In addition to the rockiness that was atypical of riparian soils, the mining had relocated some water channels that eventually flooded or buried some of the plantings. Thus, success was limited by physical circumstances. Nevertheless, the site remains fundamentally productive, as measured by

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Table 3—Height, diameter[a], and stem volume for new plantings, age 2, at Mosby Creek

		Overstory<=15%			Overstory>15%		
		Height (cm)	Diameter[a] (mm)	Volume (cm³)	Height (cm)	Diameter[a] (mm)	Volume (cm³)
Unweeded	Grand Fir	43	7.3	7.1	39	6.4	5.3
	Redcedar	57	7.7	9.6	66	7.5	10.6
	Ponderosa Pine	38	9.2	10.4	29	7.8	5.3
	White Pine	19	5.0	1.5	16	4.5	0.9
Weeded	Grand Fir	49	9.1	12.1	46	7.6	8.2
	Redcedar	55	8.2	10.9	58	6.8	8.3
	Ponderosa Pine	38	10.1	12.7	29	7.7	5.2
	White Pine	21	5.9	2.1	17	5.0	1.3

[a] Diameter refers to diameter at 15 cm above ground for grand fir, redcedar, and ponderosa pine. Diameter is root collar diameter for white pine.

its initial condition when selected. In addition to the physical limitations of this site, problems with spring and fall freezing damage occurred in both sprayed and unsprayed portions of the study.

The large Douglas-fir stock type (1+1) continued to be the largest seedlings on the units (table 1). In clearcut areas, the large Douglas-fir averaged almost three times greater volume than the small Douglas-fir. Both stock types of Douglas-fir and redcedar had less growth in underplanted areas ($p < 0.0001$). Hemlock averaged over twice the size in clearcut areas than in underplanted areas, but differences were not significant ($p = 0.7869$) in part due to low survival (14 %) in underplanted areas.

Although differences in average seedling size for Douglas-fir and redcedar in the weeded and unweeded areas were relatively small ($p > 0.0984$), competition reduced seedling growth similar to that shown for Ames Creek in figure 5, but with smaller volumes in general. Hemlock seedlings were larger in the weeded areas ($p = 0.0065$).

For the "new" plantings, at the end of two years, the grand fir and ponderosa pine were growing best (table 3). Although redcedars were the largest initially, they were exhibiting poor growth, as with the first planting. Although relative growth between the two pines was similar, the white pines were still very small, and survival was poor.

All Sites: Animal damage had a major impact on all plantations. Mountain beaver clipping occurred on three of the sites (table 4), and beaver clipping occurred on two of the sites. At least 50% of the seedlings that had been damaged by beavers were dead. Although the overall percentage of damage by beavers was low, the local impacts were severe.

As would be expected, the damage by beavers was concentrated within 15 m of streams, but damage was seen more than 30 m from the stream.

Hemlock was infrequently browsed on all sites. For the other stock types, browsing impacts were related to location of the seedling (weeded or unweeded areas, clearcut or underplanted areas), whether or not the browsing was done by deer or elk, whether browsing reduced shrub cover adjacent to the seedling, and inherent productivity of the site. Browsed Douglas-fir were reduced in stem volume at age four by 23-45 percent at the various sites. Redcedar volume reduction from browsing was greater, ranging from 50-60 percent relative to unbrowsed seedlings. As overtopping increased, the absolute impact of browsing decreased; the unbrowsed seedlings growing under high degrees of overtopping do not grow as well as those with low amounts of overtopping. The lesser impact is associated with loss of potential, over all, as long as overtopping remains.

Insect Abundance

Phase I analyses for the first and second years of response to harvest showed that stream substrate explained more variation in absolute abundance, diversity and functional feeding groups than any other stream variable ($p < 0.001$). The order in which substrate had a positive effect on total abundance was cobble>bedrock>boulder=sand= silt>gravel. Walsh (1996) provides details of this analysis by genera showing which genera were favored by each substrate medium. Of the remaining environmental variables, stream velocity explained the most variance, as would be expected from the comparatively high velocities of riffles compared with other substrates. Collector-filterers were positively associated with velocity. Mayfly (Ephemeroptera) and caddisfly (Trichoptera) genera were positively correlated

Table 4—Animal damage and mortality after 3 years (Bark Creek) or 4 years (other creeks)

|		Beaver live and dead (no.)	Beaver damaged dead	Mountain beaver live and dead (no.)	Mountain beaver damaged dead	Browsing live and dead (no.)	Browsing live	Mortality and not found (no.)	Total (no.)
		- Percent -							
Bark Creek	Large Douglas-fir	50 (8)	87	36 (5.7)	42	264 (42)	64	291 (46)	629
	Small Douglas-fir	56 (9)	61	43 (6.9)	26	273 (44)	62	264 (43)	620
	Redcedar	38 (6)	50	21 (3.3)	24	397 (63)	85	201 (32)	630
	Hemlock	19 (3)	63	19 (3)	63	25 (4)	6	305 (49)	620
Buttermilk Creek	Large Douglas-fir	77 (9.2)	86	32 (3.8)	34	225 (27)	39	423 (51)	835
	Small Douglas-fir	77 (9.3)	77	24 (2.9)	29	208 (25)	48	526 (63)	830
	Redcedar	130 (16)	43	15 (1.8)	27	27 (3.2)	5.6	457 (55)	835
	Hemlock	111 (13)	58	33 (4.0)	70	25 (3.0)	4.7	406 (49)	835
Ames Creek	Large Douglas-fir	—	—	—	—	374 (22)	27	166 (23)	721
	Small Douglas-fir	—	—	—	—	178 (25)	34	217 (30)	714
	Redcedar	—	—	—	—	193 (27)	31	108 (15)	716
	Hemlock	—	—	—	—	6 (0.8)	1	269 (38)	708
Mosby Creek	Large Douglas-fir	—	—	74 (8)	47	220 (25)	43	555 (62)	890
	Small Douglas-fir	—	—	45 (5)	51	128 (14)	32	612 (69)	890
	Redcedar	—	—	44 (5)	32	128 (14)	48	667 (75)	890
	Hemlock	—	—	25 (3)	36	24 (3)	15	777 (87)	890
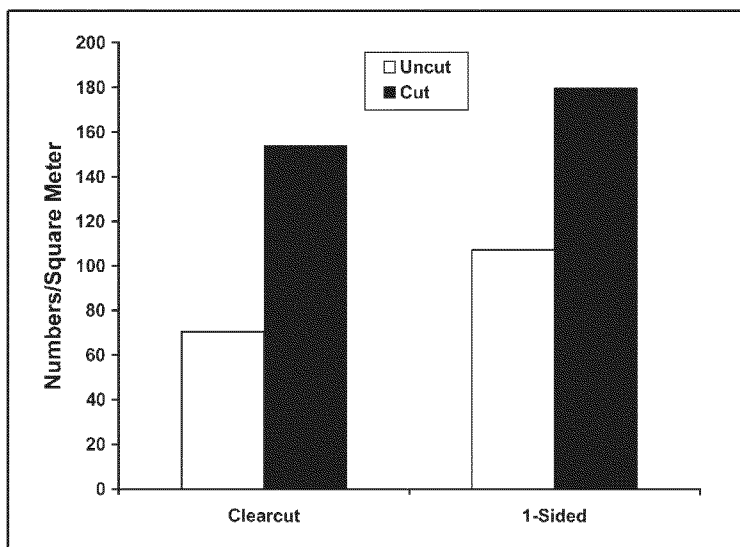
Figure 6—Total abundance of insects in each treatment type for Phase I and II studies.

with stream velocity and also negatively correlated with exposed bedrock; those which were negatively associated with velocity were positively correlated with fine-textured substrates. Radiation load, as estimated by densiometer, did not significantly explain the variance of any mayfly or caddisfly groups or functional feeding group, nor did stream temperature.

Absolute abundance of most genera increased between the first and second years following harvest ($p < 0.01$) and most genera responded positively to harvest, with the strongest response being in the second year. Only one genus, *Drumella*, displayed more abundance outside the cutting unit than within it, in general, and was more abundant both upstream and downstream from a clearing than within the clearcut. All others showing a significant response were in greater numbers within the harvest unit than above or below it ($p < 0.05$). *Cyningmula* and *Rhithrogena* were more abundant downstream from the harvest units than above the unit ($p < 0.05$).

The analyses of abundance in cut versus uncut units of Phase II data all reflected second-year responses, and they are compared here with third-year responses in Phase I experimental streams; both sets of data compare same year cut versus uncut while total abundance following clearcutting versus one-sided buffered cuts are based on two versus three years since cutting. Mean total abundance (per square meter) of benthic insects in the two uncut reaches up and downstream from each clearcut unit was half (82.6) the mean abundance in the cut reaches (165.4, $p = 0.002$, fig. 6) in both Phases I and II. Overall abundance between the

clearcut and one-sided streams did not differ ($p = 0.18$). Upstream stations had lower abundance than the two stations within cuts ($p = 0.004$) (table 5, fig. 7), and there was a general increase in insect abundance in a downstream direction, as streams increased in size. Initial conditions in streams were not identical, but numbers were comparable. Within the uncut reaches, there appeared to be fewer insects in streams scheduled for zero-buffer treatment than there were in one-sided streams, but the difference is not significant ($p = 0.132$). Abundance was similar between the two treatment types (one-sided buffers and clearcuts) in cut reaches ($p = 0.737$).

Different orders of insects displayed varying responses to harvest, but three orders were more abundant in cut than in uncut reaches. Diptera (true flies) displayed the greatest differences in mean abundance between the cut plots (17.2) and uncut plots (6.5, $p = 0.009$). Trichoptera displayed a large difference between cut (mean = 67.0) and uncut (mean = 40.7) reaches ($p = 0.025$), and Ephemeroptera also differed (22 versus 13, $p = 0.09$).

Some, but not all, effect of clearing on caddisflies and mayflies showed up a short distance downstream, where numbers were greater than upstream densities. We were unable to separate the effect of stream position from that of increased light when combining up-and downstream data for these orders with data from the clearings, but both were generally less abundant in the upstream uncut plots than in the cut plots ($p = 0.017$ and 0.09, respectively) (fig. 7). Plecoptera (stoneflies) and Coleoptera (beetles) did not differ in overall abundance between cut and uncut reaches,

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Table 5—Total abundance, insects of all genera >1.1 mm long per composite sample taken from six low-elevation western Oregon streams. Ames, Buttermilk, and Mosby Creeks were harvested both sides with no buffers in 90 or 180 m patches; Brush, Mill, and Scheele were clearcut both sides with 12 m screen south of stream for buffer on 805 m reaches

| | Upstream | Within cuts | | Downstream |
Stream	uncut	Upstream	Downstream	uncut
Ames (CC)	114	392	258	163
Buttermilk (CC)	27	115	71	2
Mosby (CC)	100	194	312	142
Percent of all insects/reach, all streams	13	37	33	17
Brush (OS)	136	140	129	131
Mill (OS)	134	163	371	137
Scheele (OS)	132	185	283	101
Percent of all insects/reach, all streams	20	24	38	18
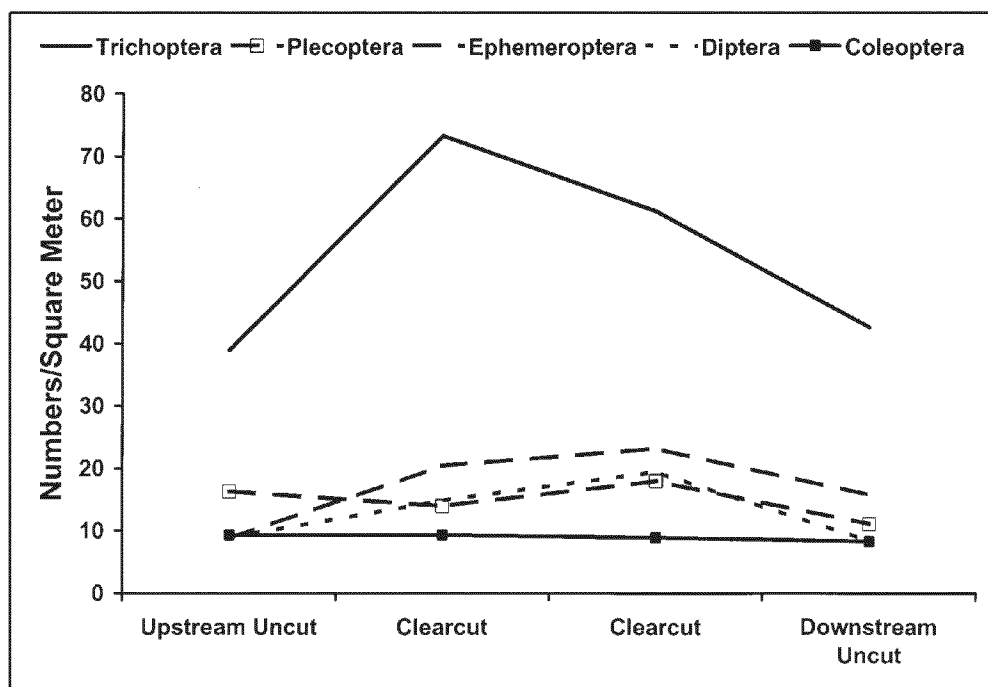
Figure 7—Total abundance of each taxonomic order for the four stations in the one-sided buffer (Phase II study).

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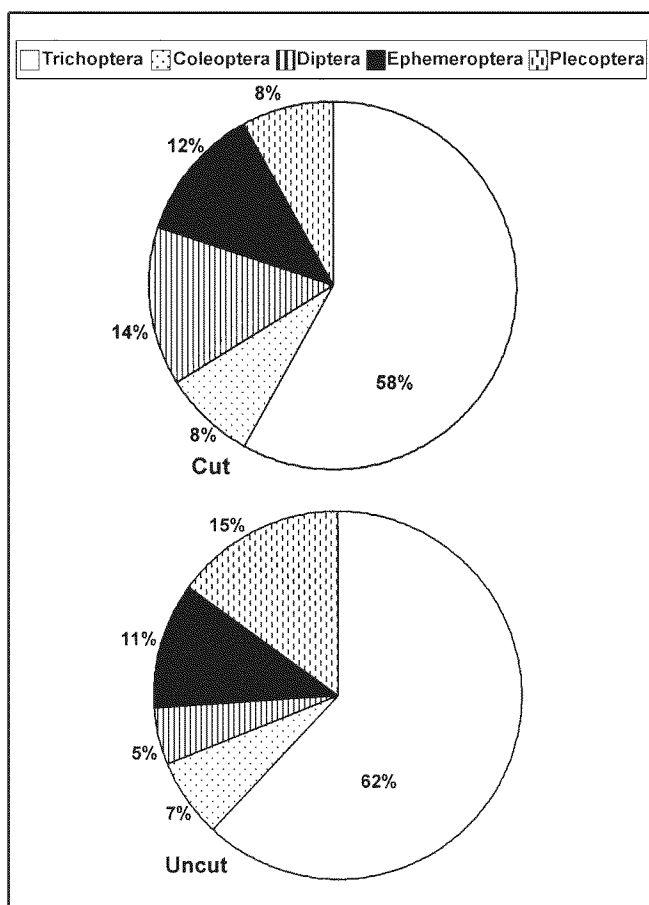
Figure 8—Relative abundance of insect taxonomic orders in cut and uncut reaches for Phase I and II studies.

hence post-harvest species ratios were somewhat altered although no order decreased with harvest.

Within cut reaches only, there appears to be 50% more Ephemeroptera and 60% fewer Coleoptera in one-sided streams than in clearcut reaches, but results were not significant ($p > 0.10$). Only five individuals of Odonata (Gomphidae, dragonflies) were collected, thus they were left out of analyses by taxonomic orders, but included in the functional feeding group analysis.

Total abundance was mildly and positively correlated with both stream discharge (r = 0.36, p = 0.013) and the seven-day maximum mean temperature (r = 0.31, p = 0.032). Plecoptera and Ephemeroptera abundance were more strongly correlated with discharge, with fewer insects when discharge was low (r = 0.49, p = 0.0004 and r = 0.51, p = 0.0002, respectively). At the genus level, *Baetidae Baetis* and *Ephemerellidae Eurylophella* in Ephemeroptera were positively correlated to discharge. No individual genera could be identified in Plecoptera.

Abundances for some insects were positively related to temperatures of streamwater. Trichoptera abundance was positively correlated with the seven-day maximum mean temperature (r = 0.53, p = 0.0001), which may explain much of the correlation for total abundance. Only two of the four genera (with adequate numbers) within Trichoptera were significantly correlated, and here the correlation was positive: *Glossosomatidae Glossosoma* (r= 0.39, p = 0.006) and *Limnephilidae Neophylax* (r= 0.41, p = 0.004). The mayflies in *Ephemerellidae Euryophella/Ephemerella* had a positive correlation (r = 0.49, p = 0.0004). No other taxonomic orders were generally correlated with temperature.

The relative taxonomic composition did not differ substantially between cut and uncut reaches of different streams (fig. 8). Only Diptera increased in relative abundance in cut stands (p = 0.03), and Plecoptera decreased slightly (p = 0.07). Relative abundance of insects between clearcut and one-sided streams differed for two insect orders. Coleoptera were more abundant in the clearcut streams (p = 0.052), both in the uncut (100% more) and cut (200% more) reaches than in the streams with one-sided buffers. The interaction was not significant (p = 0.42). Ephemeroptera were more abundant in the one-sided streams (p = 0.09), with 50% more in uncut reaches and 100% more in cut reaches, reflecting both random differences between streams independent of treatment, and also effect of treatment.

The insect genera were divided into five functional feeding groups (FFG): collector-filterers, collector-gatherers, scrapers, shredders, and predators. Some genera or families contain diverse species that belong in more than one type of FFG, thus separate categories of collector-gatherers/scrapers and collector-gatherers/shredders were included. Few genera in this sample fell into the collector-gatherer/shredder category, so those were not analyzed statistically.

Collector-filterers, collector-gathers, and predators responded similarly to harvest design (clearcut vs. one-sided buffer), and types of harvest were pooled for analysis. All functional feeding groups were more abundant in cut reaches than uncut reaches in both clearcut and one-sided streams (p = 0.009, 0.017, and 0.030, respectively). The relative abundance of insects groups did not change markedly after harvest (fig. 9). Although predators increased by almost 50%, their percentage of all insects decreased due to the overall increase in benthic insects of all kinds in cut reaches. Shredders displayed an interaction effect, with more specimens from uncut portions of one-sided streams and more in the cut portion of clearcut streams (p = 0.037). At least one of the clearcut streams (Buttermilk) had large
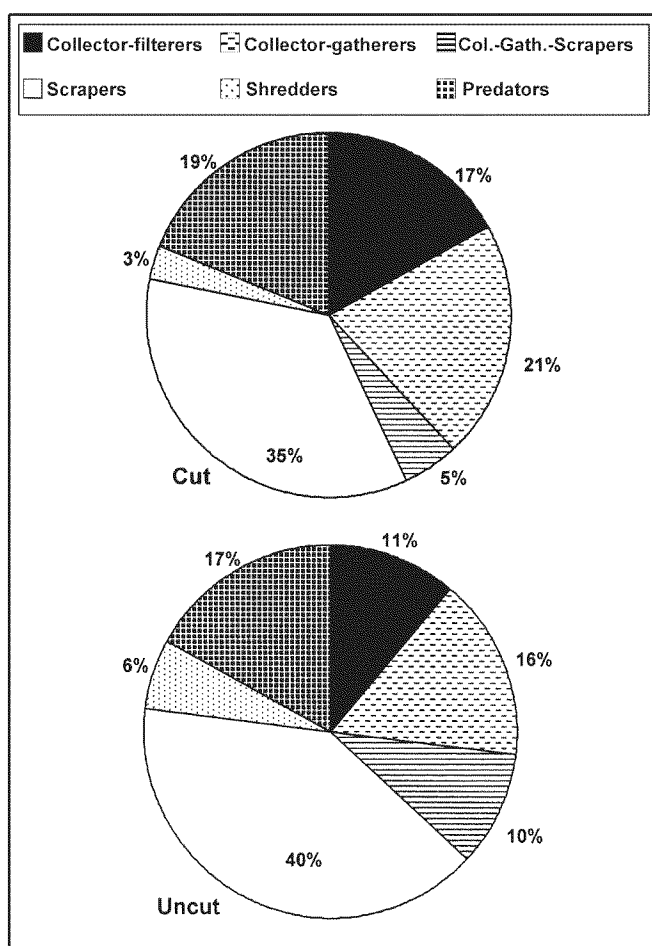
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Figure 9—Relative abundance of functional feeding groups in cut and uncut reaches for Phase I and II studies.

amounts of wood only in the cut portions, which may have trapped significant detritus and attracted more shredders.

Abundance of some FFGs varied by treatment type. Within the uncut plots there were more shredders and predators in streams with one-sided buffers, and more collector-gatherers and collector-gatherer/scrapers in the clearcut streams. Despite these apparent differences, within-cut comparisons showed only the shredders differed between treatments, with twice as many insects in the clearcuts than in the one-sided cuts. There were no differences in overall abundance between streams of the two treatment types.

The number of genera represented in each stream system did not differ by treatment types ($p = 0.743$), or between cut versus uncut ($p = 0.122$). The mean number of genera found in the uncut portions of both sets of treatment was about the same (9.3 for clearcut, 9.5 for one-side), and the means in cut reaches were also similar (10.9 for clearcut, 11.4 for

one-side). East Brush Creek (one-side) had low generic richness (2-4) at all stations in September, although the number of individuals in the samples was normal.

Stream Temperature

Temperature of water was evaluated at the warmest part of each summer, based on the moving maxima of water temperatures for seven consecutive days with the warmest mean temperatures. Temperature profiles in the four unbuffered (Phase I) experimental streams were not consistent, either in their patterns of warming or in uniformity from year to year. None appeared to warm significantly more than would be expected for a 750-m-long reach less than 8 km from the source, but lack of pre-harvest basin-wide trends were not recorded for reference. Dent (1995) indicated that there were some measurable impacts on stream temperature, but she had not compensated for the expected rate of warming in covered reaches of comparable position in the basins, a procedure regarded as necessary for estimation of net warming (Newton and Zwieniecki 1996, Zwieniecki and Newton 1999).

Streams showed variation in the amount of temperature change between clearcuts, both among streams and among years of observations, as reported by Dent (1995). The first two years of data following harvest reflected contrasts in summer temperature, with 1993 being cool and wet, and 1994 being extremely hot and dry. In 1993, the first year following harvest, water exiting the study reach ranged from 0.2°C in Bark Creek to 1.2°C in Buttermilk (main stem) warmer than water at the upper end of each study reach, based on seven-day moving mean maxima. Nearly all the net rise occurred in harvested units, as shown by Dent (1995). In 1994 during the period of least discharge, Bark Creek gained 0.6°C over all, representing gains in clearcut units and losses between cut units, while Buttermilk Creek gained about 1.6°C over all, with no consistent pattern between cut and uncut units even though the previous year had shown maximum warming to occur in harvest units. No stream gained more than 1.6°C in the length of a study reach (Dent 1995), a rise typical of the natural downstream warming trends of several headwaters streams reported by Newton and Zwieniecki (1996). Unfortunately, without baseline measurements, we are unable to determine if the observed trends represent a change from pre-harvest conditions.

The uppermost cut unit on Buttermilk Creek was actually on a major tributary rather than on the main stem. The main stem was largely slack water behind beaver dams above the confluence, hence unsuitable for study, but there was a large temperature rise at the confluence reflecting the

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warm water from the shallow beaver impoundments. This accounted for much of the total increase in the study reach. Phase II streams were described in the report by Zwieniecki and Newton (1999) which did have both pre- and post-harvest data. Their report showed that all three of the Phase II streams (sun-side buffered) were at or below pre-harvest rates of warming with downstream flow soon after water re-entered uncut stands.

DISCUSSION

The reforestation experiments demonstrate the difficulty in establishing conifers in the presence of overtopping shrubs and animals that damage seedlings. Problems with low temperature can likely be reduced or eliminated by choice of frost-hardy species, but their suitability for long-term growth in streamside environments remains untested. Herbivory will remain a serious problem regardless of species planted as long as beavers have unrestricted access to streamside regeneration.

Robison and Beschta (1990) have identified sites within 15 m of the streambank as the potential sources for most local tree-fall recruitment of large wood. Losses of regeneration attributable to beavers were most frequent in this zone, and led to elimination of most regeneration in large portions of both Bark and Buttermilk Creek experiments. Beavers were not selective among species. In these streams, beavers were not restricted in their locations by dams or other obvious residence areas, and they appeared to travel over the length of experiments and occasionally forage more than 30 m from the streams. The long-term effects of this are important, because the possibility of recruitment of conifers by natural regeneration in these areas is low owing to re-development of competing cover. Even if conifers did occasionally regenerate naturally, new seedlings would likely be damaged or killed unless the beaver pressure decreased. Fencing to keep beavers in the stream and away from the plantations worked at one site for a time. However, flooding breached the fences in several places, and cattle grazing in the units resulted in trampling of the fence. Fence failure allowed the beavers to continue foraging on the seedlings, and damage was often lethal.

Damage by deer and elk was generally not lethal, but did reduce growth. Visual examination of these plantations after ten years reveals good plantation development in clearcuts except where mortality from herbivory created large gaps. Regeneration in uncut stands was ultimately unsuccessful. The need to reduce competition for establishment and growth of conifers in riparian areas was noted by Chan et al (1998) and Emmingham et al. (2000) in their

reviews of riparian regeneration studies. They also noted that both shade tolerant and shade intolerant species were limited by overhead competitors, as we demonstrate with illustrations of interactions between canopy cover and shrubs in uncut stands, and with shrubs as overtopping where unsprayed. We also confirm that use of large, vigorous planting stock increases the likelihood of survival in both browsing and competitive environments, provided overtopping is not extreme and seedlings can be adequately planted. Howard and Newton (1984), Wagner and Radosevich (1991), and Newton et al. (1993) demonstrated a general adaptability of very large conifer seedlings in areas subject to herbivory and shrub competition in the Oregon Coast Range. Some success has also previously been observed after planting wild seedlings averaging 1.5 m tall in untreated salmonberry in shrub-dominated riparian systems near the Oregon coast (Newton, M., unpublished data, 1970, from cooperative experiments with the USDA Forest Service, Siuslaw National Forest, Waldport, OR). At Mosby Creek, the rocky soils made planting large seedlings difficult, and planting large seedlings may not be feasible in such areas.

Differences between sprayed and unsprayed areas were less than expected. Spraying with glyphosate soon after logging is typically unreliable because leaf area of shrubs has been removed to the extent that translocation to roots is sub-marginal. Application of this herbicide at least a month prior to late-summer harvest would likely have nearly eliminated shrub development except in the 3-m buffer required under Oregon Forest Practice rules for hand application. The spot treatments at Ames and Mosby Creeks were of inadequate size to be the equivalent of site preparation and were applied in spring, which led to rapid resprouting of shrubs and some damage to seedlings; treated and untreated areas were almost indistinguishable on those streams. There was also some suppression of shrubs in unsprayed parts of the experiments by herbivory, as noted in Mosby, Buttermilk and Bark Creeks where elk were abundant.

Large openings created by clearcutting demonstrated advantages in riparian management areas beyond those associated with establishment of conifers, and extend Berg's (1995) analysis beyond that associated with thinnings. The increase in benthic macroinvertebrates is consistent, both qualitatively and quantitatively with earlier reports from the Pacific Northwest. Murphy and Hall (1981) noted increased insect abundance in stream riffles within clearcuts with no residual overhead cover compared to riffles in streams dominated by old-growth stands; they observed somewhat decreased abundance in pools. Diversity was greater in clearcuts in their work, but we did not record either increases or decreases at the species level. Murphy

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et al. (1981) noted roughly the same increase in aquatic insects between uncut and clearcut reaches in summer, but also noted that differences in autumn after leaf-fall were not significant except for an increase beneath a hardwood canopy after leaf-fall. Insect numbers in riffles were consistently greater in clearcuts than in streams under canopies. They noted no differences in insect response in riffles of high- vs. low-gradient streams although there were more decreases in insect abundance in pools than increases within clearcuts, regardless of gradient. We did not determine whether there was complete compensation by autochthonous sources for decreases in allochthonous inputs of food substrates.

Favorable effects of clearcutting in terms of stream productivity and regeneration must be interpreted in the same context as reported negative consequences. Concern has been expressed with respect to temperature (Sullivan et al. 1990, Brown and Krygier 1970) to the extent that the Environmental Protection Agency recommends a temperature standard of 17.8°C (64° F) maximum for seven-day moving mean of daily maximum temperatures in streams supporting salmonids. This level is exceeded in many streams under natural conditions of forest cover (Ice et al. 2004), suggesting that silvicultural activities potentially leading to a warming trend on such streams are likely to have negative effects on stream biota. In the event clearcutting were to raise temperature to a harmful level, the practice would not be permitted; under Oregon, Washington and California laws, clearcutting to the stream is either prohibited or tolerated only in rehabilitation situations regardless of baseline temperature patterns. The question remains open as to whether this limits future net stream productivity by restricting creation of openings with positive long-term effects.

Our streams did, on occasion, exceed 17.8°C briefly. We have inadequate data to determine whether downstream temperature was consistently raised or not raised as the result of cutting, and data reported by Zwieniecki and Newton (1999) suggest that increases in uncut areas downstream were in a range to be expected high in headwaters basins. In no case did reported temperatures rise enough to threaten mortality or cause a persistent depression of metabolism of fish or macroinvertebrates, according to current literature (Sullivan et al. 2000). Beschta et al. (1987) and Sullivan et al. (2000) have pointed out that fish can tolerate short-term exposure to temperatures several degrees higher than optimum without injury. Their reviews pointed out that the higher temperatures slow feeding rates while metabolism is elevated in the warmer conditions, and that this results in slower weight gain. Brett et al. (1969) describe maximum growth as increasing with temperature up to 15°C so long

as food supplies are adequate. His data also indicate that growth of sockeye salmon fingerlings is positive (although not maximum) at temperatures well above 20°C. Data, largely from laboratory studies, has shown that tolerance to elevated temperatures is inversely related to duration or exposure to elevated temperatures (Sullivan et al. 2000). They reported that salmon can spend lengthy periods above 24°C without mortality or loss of vigor on return to cool conditions. The temperature effects reported above were from laboratory studies with temperatures held at constant levels. McMahon et al. (2001) exposed bull trout and brook trout to constant versus temperatures fluctuating plus or minus 3°C with the same mean temperatures. They observed that fish growth was slightly greater when water temperature was constant than when mean water temperature was the same but fluctuating, reflecting that less time is spent within the optimum range. These data indicate that it may be possible for fish populations to tolerate some exposure to temperatures above 17.8°C, but the impacts of fluctuating natural streams, repeated occurrences of such peaks for several days, and duration of exposure under different levels of food supply are unknown.

Inferences from this study pertaining to water temperature from the reforested reaches were limited in their ability to determine specifically where warming took place under variable canopies. Dent (1995) did show that most warming occurred in complete clearings and that some warming but more cooling occurred in shaded reaches. Data gathered in the sun-sided buffered streams were more comprehensive. None of the sun-sided buffers except Scheele Creek resulted in a net rise in temperature within the cutting unit despite the lack of buffer on one side. Scheele Creek showed slight warming in the lower end of the treated reach, but that peak was not observed immediately downstream from the cut unit. The data from Zwieniecki and Newton (1999) from clearcuts 16 to 48 ha in size indicated that average maximum temperatures on 16 streams may or may not rise above baseline maximum temperatures within harvest units. Their highest seven-day moving mean maximum temperature 300 m downstream from the harvest units never exceeded 2°C more than baseline trends, and based on one year of baseline and one year post-harvest data, there were as many streams cooler 300 m downstream from units as there were streams warmer than the baseline trends. These data demonstrated the pitfalls of reliance on single-year observations, of looking only in clearcuts and immediately upstream areas, and of failure to adjust for the expected downstream temperature rise before cutting.

The "sun-sided" buffers accomplished what was predicted for them by Newton's (1993) report regarding silvi-

cultural prescriptions in riparian zones. The sun-screen on the south side intercepted most incoming direct radiation, and also presumably allowed outgoing radiation. Related work on streams in California (Newton and Zwieniecki 1997) has shown that streams flowing with no cover whatever, as in large burns, fluctuate greatly in diurnal temperature. In those streams, warming during the day appeared to be attenuated by streams becoming very cold at night, presumably reducing the stored heat in sediments and rocks so as to reduce heating during the day in the direct sun.

When streams absorb heat, as in a fully exposed clearcut reach on a hot day, peak temperatures tend to decrease when they re-enter a covered reach, but in an uneven pattern. The degree to which cooling may occur downstream from cutting units in the cooler air provided by shade is variable, depending on the individual stream system. It appears that over all, stream temperature is determined by a downstream temperature fluctuation pattern, or "signature" comprised of combinations of discharge, velocity, points of cool-water baseflow and confluences, coupled with patterns of gaps in cover (direct radiation) and air temperature patterns with decreasing altitude. Year-to-year variation in air temperature could be expected to raise or lower the whole trend. Radiation level does not appreciably vary from year to year except through changes in interception of radiation attributable to changes in plant cover. The combination of increased radiation and elevated air temperature on fresh slash adjacent to the stream would lead to energy loading on the stream that would increase with regional air temperature. Of significance is that the "sun-sided" buffers permitted the streams to stay within their signature patterns with no persistent hot spots, as nearly as we can determine, and are favorable to both regeneration and benthic insect productivity.

We have not addressed other impacts of clearcutting to streams. These may include bank damage, loss of existing trees with their woody debris recruitment potential, and loss of sources of leaf-fall and other inputs of organic debris used as food for aquatic invertebrates. We have also not addressed whether increases in insect abundance affect fish productivity and whether clearings affect insect abundance and stream temperature at other times of the year. We close with the caveat that logging needs to be done with care to avoid soil and streambank disturbance. It also is likely advantageous to leave strips of shrubs several meters wide along the banks both for shade and litter, regardless of other features of buffer design.

CONCLUSIONS

1. All three principal species of conifer—Douglas-fir, western hemlock, and western redcedar—displayed potential for good growth in riparian environments. Large seedlings grew better than small seedlings. Seedlings of ponderosa pine and grand fir were frost-hardy, but we have no measure of their long-term suitability for these types of plantings.

2. For satisfactory seedling growth of all species, both residual hardwoods and overtopping shrubs should be kept at a low level, however, a screen of shrubs should be retained along the bank to shade the stream and provide litter inputs.

3. Cover removal to the extent done in these experiments (Phases I and II) did not appear to cause warming of water by 300 m downstream of harvest units. In a companion study, none of the streams gained or lost more than 2°C compared to pre-harvest conditions when measured in uncut units 300 m downstream of cut units, based on the seven-day moving mean maximum for the seven consecutive warmest days. Clearcutting to the streambank facilitated regeneration and increased benthic productivity.

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SLENDERNESS COEFFICIENT IS LINKED TO CROWN SHYNESS AND STEM HYDRAULICS IN LODGEPOLE PINE

Victor J. Lieffers[1] and Uldis Silins[1]

ABSTRACT

This paper outlines several studies on hydraulic limitations, crown size and leaf area development in lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.). A central theme of this work is that the bole slenderness coefficient (height/diameter ratio) is linked to productivity by means of two mechanisms. First, trees with slender boles are less stiff and thus are more likely to move widely in wind. In taller stands, wide movement of trees in windstorm will result in violent collisions with neighboring trees; this results in crown abrasion and narrow crowns with unoccupied space between crowns, i.e., crown shyness. Stout trees, in contrast, have only small oscillations in wind and neighboring trees often have overlapping branches. Second, slender trees develop narrow annual rings with reduce permeability of the wood for transport of water to crowns. Also, the reduced strength of slender trees will results in increased bending of the bole, which is also likely to reduce the permeability of the xylem. Bole slenderness, therefore, likely plays a role in stand productivity, the ability of trees to release following thinning, and in the probability of trees remaining alive after canopy disturbance late in stand development. Managers can manipulate bole slenderness by stand density management.

KEYWORDS: Lodgepole pine, thinning, crown slenderness, height/diameter ratio, xylem permeability.

SUMMARY[2]

Slenderness coefficient, i.e., the height/diameter ratio of a tree, has long been linked to the biomechanical relationships of trees and wind stability (Ruel 1995). Trees with stout stems will resist deflection in wind much more than slender stems (fig. 1). This short paper briefly outlines how the slenderness coefficient of trees may be important to the development and maintenance of crown size, crown leaf area (crown shyness) and the hydraulic supply of that leaf area. Examples are given for lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.).

Crown shyness is the empty space that develops around individual crowns of trees in fully stocked, maturing stands (Putz et al. 1984) (fig. 2). Some researchers believe that the empty space develops as a result of shading of foliage between crowns (Umeki 1995; Cescatti 1997), but in reality there is usually less leaf area in older stands (Ryan et al. 1997) than in young stands, where there is little crown shyness. In contrast, there is increasing evidence that the space between crowns develops as a result of abrasion of the crown during movement of trees during wind (Long and Smith 1992; Smith and Long 2001, Rudnicki et al. 2001). Indeed, the crown of a 15-m tall lodgepole pine tree may collide with neighboring trees more than 40 times per minute when wind speed averages 5 m/s (Rudnicki et al. 2002). Similarly, crowns may oscillate more than 6 m in a wind gust (Rudnicki et al. 2001), and reach velocities of > 7m/s (Rudnicki et al. 2002). Impacts in these conditions would therefore have sufficient energy to cause serious damage to twigs, and branches (Grier 1988) especially in
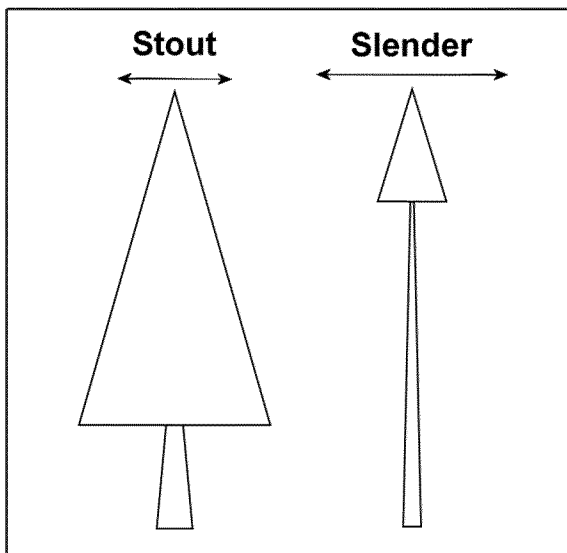
Figure1—Deflection from a gust of wind for trees of the same height with slender and stout boles.

Figure 2—A fully stocked, 16-m tall lodgepole pine stand in the Yukon Territory. The photograph was taken at 1.7 m above the ground with a 55-m lens.

cold weather where twigs are very brittle (Lieffers et al. 2001). As bole stiffness and the ability of the stem to resist deflection is strongly related to bole diameter (Silins et al. 2001), stems with low slenderness coefficient are expected to move less in wind storms than slender stems. As hypothesized, stands with slender stems had much wider movement of crowns than those with stout boles of the same height (Rudnicki et al. 2002). In a second study, crown shyness was estimated in small plots in wild stands of lodgepole pine (Rudnicki et al. 2004). When stems were less than about 12 m tall, there was little crown shyness in fully stocked stands; in these short stands crown closure was apparently driven by relative density. In stands > 12 m high however, there was increasing crown shyness in plots with trees of high slenderness coefficient.

Water relations of trees are also likely to be related to the slenderness coefficient. First, slender trees have narrow annual rings. Sapwood of lodgepole pine with rings < 0.5 mm has reduced permeability to water flow compared to sapwood with wide annual rings (Reid et al. 2004). Reduced permeability to water flow is expected to become a limiting factor in transport of water to foliage in crowns, thereby limiting growth. Secondly, there is increasing evidence that bending deflection of trees by wind damages the permeability of tree stems (Frederickson et al. 1994, Silins et al. unpublished). In a study of juvenile lodgepole pine following thinning, stems were either tethered to reduce sway or

had sails added to increase deflection in wind (Liu et al. 2003). Sapwood permeability after thinning was related to sway treatments; sapwood permeability was ranked control (unthinned) > tethered (thinned) > untethered (thinned) > sail added (thinned).

Since managers can use stand density management to manipulate the bole slenderness coefficient, we hypothesize that a thinning regime where stems are kept at slightly lower relative density than typically develops in self-thinning stands will produce stems with stiffer boles that are more resistant to deflection. Stems that flex less will maintain larger crowns in the second half of the rotation. Stems that resist bending will also suffer less damage to xylem sapwood during strong wind events and maintain their ability to deliver water to crowns. Finally lodgepole pine stems with wider rings (often produced after stand spacing) have higher sapwood permeability.

ACKNOWLEDGEMENTS

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